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Using stand replacement fires to restore southern Appalachian pine–hardwood ecosystems: effects on mass, carbon, and nutrient pools

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Abstract

Pine–hardwood ecosystems in the southern Appalachians are in serious decline due to fire exclusion and insect infestations. Fire has been advanced as a tool to restore these ecosystems, yet there are few studies evaluating overall ecosystem effects. Our objectives were to evaluate the effects of stand restoration burning on forest floor nitrogen (N) and carbon (C) pools, and soil and stream chemistry. We measured changes in forest floor (coarse woody debris, small wood, litter, and humus) mass, N, and C; changes in soil chemistry (calcium (Ca), potassium (K), magnesium (Mg), cation exchange capacity (CEC), pH, C, and N); and changes in stream nitrate (NO_3). Results showed that significant reductions in mass, N, and C occurred only for litter and small wood on the ridge, where N losses were 52.9 kg ha^{-1} for litter and small wood combined. No significant effects were observed on the mid- or lower slope of the treatment watershed. Losses on the ridge are considerably lower than losses which occur with alternative burning treatments used in the region, such as the fell and burn treatment. Soil and stream chemistry showed no response to burning. Spatial heterogeneity in fire intensity (combustion temperatures ranged from <52–>800°C) and severity associated with stand replacement burning results in a mosaic of fire effects and considerably less consumption and subsequent nutrient losses.

1. Introduction

Forests of the southern Appalachian region are undergoing considerable change. The current composition, structure, and the trajectory of future conditions of these forests has largely been shaped by historical and current anthropogenic and natural disturbances. For example, Native Americans burned extensive forested land areas for agriculture and hunting for 10 000–12 000 years (DeVivo, 1991). Beginning in the mid-1800s, European settlers also used fire, in combination with land clearing, and nearly the entire southern Appalachian region was logged during the early 1900s (Stephenson et al., 1993). About the same time, chestnut blight (*Chryphonectria parasitica* Murr.) decimated American chestnut (*Castenea dentata*) populations, which formally occupied approximately 35% of the basal area of the oak–hickory forest-type (Woods and Shanks, 1959). Subsequently, fire exclusion, smaller scale logging, and reversion of agricultural land to forest have further shaped forests in the region.

One of the most heavily impacted ecosystem-types is the pine–hardwood ecosystem which requires disturbance for both its establishment and maintenance. Pine–hardwood ecosystems typically occupy the most xeric sites (i.e. south/west aspects) and consist of varying proportions of pitch pine (*Pinus rigida* Mill.),

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virginia pine (*Pinus virginiana* Mill.), and/or shortleaf pine (*Pinus echinata* Mill.) and a mixture of hardwoods, including scarlet oak (*Quercus coccinea* Muenchh.), chestnut oak (*Quercus prinus* L.), and red maple (*Acer rubrum* L.). Mountain laurel (*Kalmia latifolia* L.), an evergreen ericaceous shrub, is a major component of these ecosystems. Many of these stands are the result of past agricultural activities which created microsite conditions conducive to pine regeneration (i.e. mineral soil, limited competition) (Whittaker, 1956; Nicholas and White, 1984). However, others are located on sites which could not be cultivated due to steep topography and poor soils and fire has been advanced as a major factor determining their origin. In either case, the maintenance of pine–hardwood ecosystems is thought to depend on intense wildfires (Barden and Woods, 1976). As pine–hardwood sites are typically dry, hot, and contain substantial quantities of flammable fuels (Vose and Swank, 1993), natural or human-caused fires have the potential for the high fire intensity necessary for pine regeneration (Barden and Woods, 1976).

The pine component of most of these pine–hardwood ecosystems is in serious decline. Smith (1991) determined that 98% of the land area occupied by this type has little or no remaining live pine. Smith's study showed that pine has been declining since the early 1970s; however, a major loss of pine occurred in the mid-1980s. This loss is coincident with a severe drought in the region (Swift et al., 1989) which weakened trees and caused widespread southern pine beetle (*Dendroctonus frontalis* Zimm.) infestations. Shifts in land use and fire suppression have limited the role of either human-caused or natural disturbances in the establishment and/or maintenance of these ecosystems. Although fuel loads are currently substantial (due to pine mortality and dense mountain laurel), suppression efforts will continue to prevent large-scale wildfires, even during dry conditions. As an alternative, silvicultural treatments will be required to restore and maintain the pine–hardwood ecosystem-type within the southern Appalachians.

Over the past 20 years, some of these degraded pine–hardwood stands have been chainsaw-felled, burned, (i.e., 'fell and burn') and planted to white pine (*Pinus strobus* L.) in an attempt to increase overall site productivity (Swift et al., 1993). Studies have shown that this treatment also increases the density of other pine species more typical of the site (i.e. *P. rigida, P. virginiana, and P. echinata*) (Vose et al., 1994; Clinton et al., 1993). More recently, stand replacement fires have also been prescribed because of the high costs of the 'fell and burn' treatment (~\$500 ha^{-1}) and concerns over large losses of site nitrogen (N) (Vose and Swank, 1993). In this application, degraded stands are burned with the objective of creating seedbed conditions for pine seed germination and reducing mountain laurel vigor to allow for seedling establishment. An alternative to the fell and burn technique is stand restoration burning; that is, fires are ignited in intact stands (i.e. trees and shrubs are not chainsaw-felled) and no planting is conducted after the fire. The stand restoration prescription reported in this study also had an additional objective of stimulating forage production and oak regeneration in the mid-slope and lower-slope positions of the treatment watershed. Previous studies have reported the success of these alternative treatments on pine regeneration (Vose et al., 1994; Major, 1996; Elliott et al., 1997) and other vegetation responses (Clinton et al., 1993; Elliott et al., 1997). In addition, it is equally important to evaluate effects on other ecosystem components to assess whole system responses. Effects on biogeochemical cycling are key indicators, because changes in nutrient pools and cycling rates directly impact short- and long-term site productivity. For example, treatments which stimulate pine regeneration, but substantially reduce long-term site productivity, may not be acceptable restoration alternatives. Changes in N are especially important, because N most commonly limits forest productivity (Vitousek et al., 1982), many of these sites have inherently low N availability (Knoepp and Swank, 1993), and N losses can be substantial because of low volatilization temperature (i.e. 200°C) (Boerner, 1982).

The objectives of our study are to evaluate the immediate effects of stand replacement burning on forest floor carbon (C) and N pools, soil nutrients, and stream chemistry. We focus on these three components (i.e. forest floor, soil, and streams) of the biogeochemical cycle because they are the primary factors determining site productivity (forest floor and soils) or, as in the case of stream nutrients, are important indicators of altered biogeochemical cycles. For example, in southern Appalachian hardwood ecosystems, approximately 50% of the total available soil N is provided by the forest floor (Monk and Day, 1988). Similarly, short-term changes in soil nutrients due to mineral release and downward movement of volatilized N (Klopatek et al., 1990; Covington et al., 1991) can have an immediate effect on regeneration success, and changes in pH, soil C, and CEC could have longer term effects on site productivity (DeBano, 1991) via changes in nutrient cycling rates. Due to the conservative nature of forest ecosystems in the southern Appalachians, stream N has been used as an indicator of disturbance effects on N cycling rates (Swank et al., 1988), and as an integrated measure of ecosystem responses to fire (Tiedemann et al., 1979).

2. Methods

2.1. Site description

The study area is located on the Nantahala National Forest in the southern Appalachian region of western North Carolina and is part of the Wine Spring Creek Ecosystem Management Project. The study site is a 82 ha, steep (40%) south-facing slope. Elevation varies from 1200 m near the stream to 1500 m on the ridge. Mean annual temperature is 10.4°C and mean annual precipitation is 1900 mm. Soils near the ridge are Typic and Lithic Dystrochrepts (Edneyville and Cleveland series) and soils in mid- and lower slope are Typic Haplumbrepts (Cullasaja series). Overstory species composition varied considerably from the lower and mid-slope to the ridge of the study area, whereas basal area was comparable ($\sim$28 m^2 ha^{-1}) (Elliott et al., 1997). Major overstory species were *P. rigida*, *Q. prinus*, and *A. rubrum* on the ridge; and *A. rubrum*, *Carya* spp., *Q. coccinea*, and *Tsuga canadensis* on the mid- and lower slope.

2.2. Plot layout

Plots were arrayed along four transects which extended from the ridge to the stream on three transects (A, B, and D), and only on the ridge position on transect C (a steep cliff prevented extension of transect C to the stream) (Fig. 1). Plots were 10×10 m^2, and were located on the ridge (n=8), mid-slope (n=6), and lower slope positions (n=6). Within each topographic position and transect, plots were randomly located on either the east or west side of the transect.

2.3. Fire characterization

The study site was burned on 28 April 1995 using a helicopter and helitorch. Streams on the east side and bottom of the study area provided natural fire breaks. An access road and a back fire were used as fire breaks on the west side, and a back fire was set along the top of the ridge. Back fires did not burn any of the study plots. The remainder of the study area was ignited with strip head fires along the lower and mid-slope. Fire in the lower slope positions was spotty and confined to the understory. The fire crowned frequently in both the upper mid-slope and ridge positions.

To quantify combustion temperature at each plot, we used heat-sensitive paint (Omega Engineering) applied to ceramic tiles (10×20 cm). One to 2 days prior to burning, tiles were located at 1 and 2 m aboveground on either the East or West side of each measurement plot. Heat sensitivities ranged from 52 to 804°C, at increments of approximately 20°C between 52 and 400°C, and 100°C between 400 and 804°C. The day after the fire, tiles were returned to the laboratory and examined. Melted paint indicated that the temperature threshold for a given paint was exceeded. As such, we used the temperature tiles to bracket combustion temperatures for each plot (e.g., 700–800°C).

Heat penetration into the forest floor was determined using heat-sensitive paint applied to long and narrow ceramic tiles (i.e. 2.54×12 cm). The paints melt at 45 and 59°C, a range that brackets the thermal lethal point for most plants (Hare, 1961). Tiles were inserted through the forest floor and into the mineral soil at three random locations on each plot. After burning, tiles were relocated, returned to the lab, and the depth of heat penetration determined. Values from the three tiles were averaged to provide an overall estimate heat penetration for each plot.

2.4. Forest floor mass, C, and N pools

Downed wood $\geq$7.5 cm diameter (coarse woody debris or CWD) was sampled on each 100 m^2 plot prior to and after burning. Pre-burn CWD volume was estimated by measuring total length and circumference at 1 m intervals. Circumference was also measured at the ends of the logs or at the point where the log crossed the plot boundary. Volume was computed

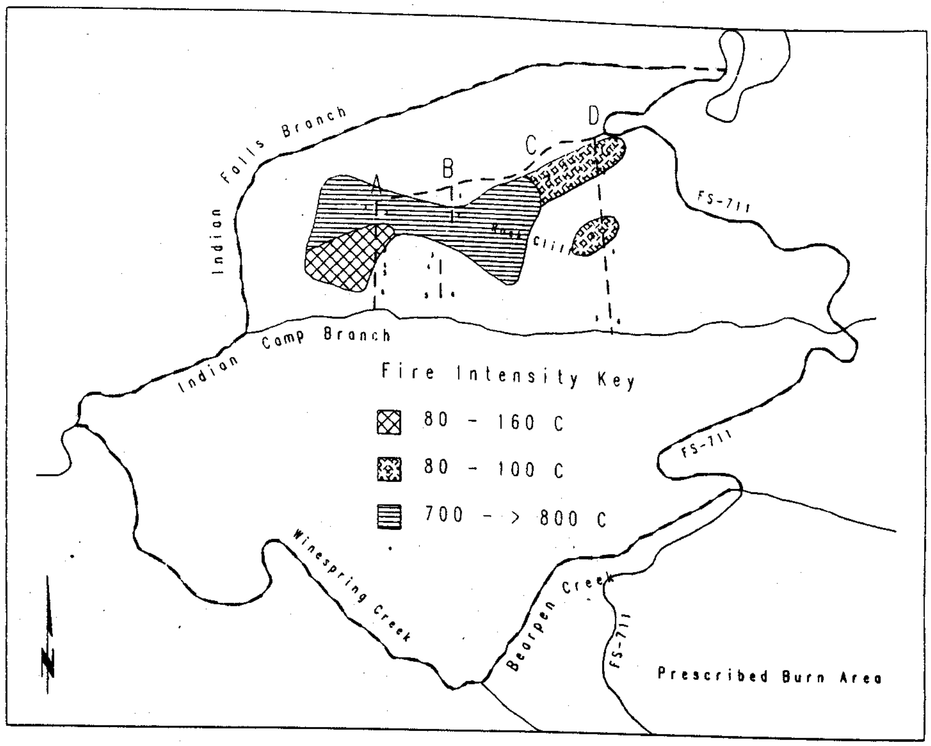

Fig. 1. Transect (A, B, C, and D), plot locations (1–6 along dashed transect lines), and fire intensity map for the burn study area.

for each 1 m segment, assuming uniform taper between the circumference measurement points. Volume was summed for all logs and log sections within a plot. In addition, wire bands were wrapped tightly around the logs at the locations where circumference was measured. CWD was also categorized by decay classes ranging from 1 to 4, with a value of 1 representing freshly fallen logs and 4 representing highly advanced decay. Volume was converted to mass using specific gravity (SG in g cm^{-3}) estimates based on wood samples taken from each measured log (Table 1). To estimate mass consumption by the fire, wire bands were re-tightened around the log and post-burn circumference determined. Volume was recalculated and converted to mass using pre-burn procedures. Mass loss was estimated by subtracting pre-burn CWD mass from post-burn CWD mass for each plot. Pre- and post-burn C and N pools for CWD were estimated using C and N concentrations for similar sites and species (Vose and Swank, 1993).

Table 1
Pre-burn mean specific gravity by decay class for CWD (>7.5 cm)

Decay class	Specific gravity
1	0.461 (0.030)
2	0.525 (0.070)
3	0.383 (0.035)
4	0.356 (0.038)

Values in parentheses are standard errors.

Prior to burning, other forest floor components were sampled using four 0.09 m^2 subplots randomly located within each 100 m^2 plot. Sub-plot corners were marked with a metal pin flag. Material within the subplots was separated into three components: small wood less than 7.5 cm diameter, litter (Oi), and a combined fermentation and humus component (Oe+Oa). A 0.3×0.3 m^2 wooden sampling frame was used to define the sampling area. Small wood (<7.5 cm diameter) within the sampling frame was cut using pruning shears, and forest floor was removed by component (i.e. Oi, Oe+Oa) after cutting along the inside of the sampling frame with a sharp knife. Within 1 week after burning, the forest floor was re-sampled with the procedures used for the pre-burn sampling. Sub-plot corners were re-located, and the post-burn samples were taken in a random location within 1 m of the pre-burn subplot. After both pre- and post-burn sampling, forest floor components were bagged separately and transported to the laboratory where they were dried for at least 72 h at 60°C and weighed.

To determine C and N of forest floor components at the plot level, sub-plot samples were composited, ground, and analyzed for C and N concentration with a Perkin Elmer 2400 CHN analyzer.

2.5. Soil chemistry

One composite soil sample was collected from the surface 0–5 cm of mineral soil for each plot prior to the burn (March 1995) and 3 months after the burn (July 1995). Each composite consisted of 18–24 individual samples taken across the 100 m^2 plots, which were collected with an Oakfield soil sampler (1.8 cm inside diameter). Soils were air-dried before analysis. All soil data presented are on an air-dry soil basis.

Soil pH was measured with 10 g of soil in a 1:1 soil:0.01 M $CaCl_2$ solution. Soil cation concentrations were determined by extraction with a vacuum soil extractor (Centurion International, Lincoln, NE). A 10 g subsample was placed in a mechanical vacuum extractor and extracted with 50 ml of 1 M NH_4Cl for 12 h. Excess NH_4Cl was removed by washing with 95% EtOH for 1 h. The NH_4 remaining on cation exchange sites was then leached from the soil using 1 M KCl solution for determination of cation exchange capacity. NH_4Cl solution was then analyzed for Ca, Mg, and K using a PE 2100 Atomic Absorption Spectrometer. KCl solution was analyzed for NH_4^+ content with a Perstorp Autoanalyzer using the alkaline phenol method (Technicon Instruments Method AA2 98-70W, Technicon Instruments, Terrytown, NY). Total C and N were determined by combustion using a PE 2400 CHN Analyzer.

2.6. Stream chemistry

A network of six stream sampling sites was established in February, 1993 to characterize dissolved inorganic chemistry in the Wine Spring Creek Basin. Chemistry sampling was co-located with sites established for suspended solid sampling; sites were strategically distributed across the drainage system to evaluate responses to stand restoration burning and silvicultural prescriptions in other parts of the Wine Spring Creek study area. Grab samples were collected weekly from each site and select storms were sampled using time increment-proportional samplers. Samples were analyzed for pH, HCO_3^-, NO_3^-, NH_4^+, Cl^-, K^+, Na^+, Ca^{2+}, Mg^{2+}, SO_4^{2-}, and SiO_2 using protocols and established methods at the Coweeta Hydrologic Laboratory (Deal et al., 1996). The effects of burning on stream solute concentrations were assessed using pre- and post-burn concentration regression analyses for sites draining the burned area versus unburned sites (reference areas).

2.7. Statistical analyses

The effects of burning (i.e. pre-burn vs. post-burn) on soil nutrients and aboveground mass, N, and C pools were analyzed using paired t-tests (PROC MEANS, SAS Institute Inc., Cary, NC). Differences among slope positions (pre- and post-burn analyzed separately) were tested with ANOVA and a protected least significant difference test (LSD) (PROC GLM, SAS Institute Inc., Cary, NC). Significance levels were set at $p<0.10$ for all tests.

3. Results

3.1. Combustion temperature and forest floor heat pulse

Flame temperature (i.e. at 1 and 2 m aboveground) varied considerably across slope positions (Table 2). On the lower slope, temperature never exceeded 52°C (the lowest detection limit of the heat-sensitive paint). On the mid-slope, maximum temperature was 160°C and minimum temperature was less than 152°C. Combustion temperature was greatest on the ridge plots where temperature exceeded 804°C on several plots. This was not uniform over the entire ridge, however, as a few of the ridge plots burned at substantially lower combustion temperatures (i.e. <52°C).

Heat penetration into the forest floor also varied considerably across slope positions, with a pattern of

Table 2
Variation in flame temperature and heat penetration by slope position, transect location (flame temperature), and height above ground (flame temperature)

Slope position	Flame temperature (°C)								Heat penetration (mm)	
	Transect A		Transect B		Transect C		Transect D			
	1 m	2 m	1 m	2 m	1 m	2 m	1 m	2 m	45°C	59°C
Ridge	>800	>800	<52–800	<52–800	>800	>800	<52–107	<52–90	27.5 (9–55)	24.0 (7–55)
Mid-slope	93–160	80–90	<52	<52	n.a.	n.a.	<52–90	<52–90	18.2 (1–52)	16.8 (4–52)
Lower slope	<52	<52	<52	<52	n.a.	n.a.	<52	<52	0.5 (0–3)0.6	0.6 (0–4)

Values are ranges for flame temperature and mean values for heat penetration with ranges in parentheses.

response consistent with combustion temperature (Table 2). On the ridge, a 45°C heat pulse penetrated 27.5 mm into the forest floor and a 59°C heat pulse penetrated 24.0 mm. On the mid-slope, a heat pulse penetrated 18.2 and 16.8 mm for 45 and 59°C, respectively. On the lower slope, a heat pulse penetrated less than 0.6 mm. Soil temperature never exceeded 45°C on any of the plots.

3.2. Forest floor mass, C, and N pools

Pre-burn forest floor mass varied considerably by component and slope position. Averaged across slope positions, humus had the greatest mass (18 809 kg ha^{-1}), varying from 11 038 in the lower slope to 30 609 kg ha^{-1} on the ridge (difference significant at $p<0.10$; Table 3). Fire significantly reduced small wood and litter C on the ridge (by 78 and 65%, respectively). Wood and litter C were reduced by 41 and 25%, respectively, on the mid-slope, but as with mass, these changes were not statistically significant. After burning, litter C was significantly greater in the lower slope than on the ridge (Table 3).

Averaged across slope positions, pre-burn N pools were greatest in the humus (271 kg N ha^{-1}), followed by litter (43 kg N ha^{-1}), small wood (26 kg N ha^{-1}), and CWD (17 kg N ha^{-1}) (Table 3). Variation in pool size was a result of both differences in mass and N concentration. For example, N concentration varied considerably among components, with greatest values for humus (1.48%), followed by litter (0.97%), and small wood (0.55%) (Table 4). Small wood, litter, and humus N pools were greatest in the ridge, while N pools in CWD were greatest on the lower slope. This variation was largely a result of differences in component mass among slope positions. Fire resulted in significant N losses from small wood and litter pools on the ridge position, with losses of 28.5 and 24.4 kg N ha^{-1} for wood and litter, respectively. Effects of fire on N pools were not statistically significant for all other slope positions and components. After burning, N concentration significantly increased for litter in the mid-slope with no differences among other slope positions (Table 4).

3.3. Soil and stream chemistry

Prior to burning, slope position had a significant effect on soil chemical properties including concentrations of K and Mg, CEC, pH, and percent total C and N (Fig. 2). In general, values of all chemical constituents were lower on the ridge than on other slope positions, although differences were not always significant. Significant differences ($p<0.10$) between pre-burn and post-burn sampling were observed for CEC in the ridge, and K, Mg, CEC, and pH in the mid-slope. With the exception of pH, post-burn values were lower than pre-burn.

Stream chemistry in the WSC basin is characterized by low concentrations (generally <1 mg l^{-1}) of most solutes (Table 5) and is quite similar across all sites measured in the basin. Moreover, concentrations of most constituents are within the range of long-term mean values for streams draining baseline watersheds at the Coweeta Hydrologic Laboratory (Swank and Waide, 1988) except for NO_3^- and SO_4^{2-}; in both cases WSC values are higher. Examination of stream NO_3^- concentration for a 29-month period after the burning treatment showed that the stand restoration burn had no measurable effects on $NO_3^- - N$ or $NH_4^+ - N$. For

Table 3
Pre-burn and post-burn mass, N, and C pools (all in kg ha^{-1})

	Mass			C			N		
	Pre	Post	Loss	Pre	Post	Loss	Pre	Post	Loss
Ridge									
CWD	8 776(2830)^{a}	7 726(2800)^{a}	1050	4 178(1347)^{a}	3 678(1333)^{a}	500	13.8(4.4)^{a}	12.1(4.4)^{a}	1.7
Small wood	6 933(2107)^{a}	1 369(770)^{a}	5564*	3 441(1051)^{a}	735(406)^{a}	2706*	36.4(13.2)^{a}	7.9(4.3)^{a}	28.5*
Litter	5 362(365)^{a}	1 873(438)^{a}	3489*	2 726(201)^{a}	946(336)^{a}	1780*	51.9(4.4)^{a}	27.5(12.3)^{a}	24.4*
Humus	30 609(7906)^{a}	28 449(77.83)^{a}	2160	13 646(4010)^{a}	13 154(3803)^{a}	492	436.7(112.5)^{a}	413.4(102.6)^{a}	23.3
Total	51 680	39 417	12 263	23 991	18 513	5 478	538.9	460.9	77.9
Mid-slope									
CWD	8096(3036)^{a}	7 308(2675)^{a}	788	3 855(1445)^{a}	3 478(1273)^{a}	377	12.7(4.8)^{a}	11.5(4.2)^{a}	1.2
Small wood	4234(1295)^{a}	2 465(492)^{a}	1769	2 121(655)^{a}	1 242(238)^{a}	879	23.4(7.3)^{a}	14.2(2.7)^{a}	9.2
Litter	3 775(660)^{b}	2 825(737)^{ab}	950	1 841(338)^{b}	1 384(355)^{ab}	457	32.6(5.8)^{b}	36.2(10.4)^{a}	−3.6
Humus	14 780(5157)^{ab}	13 849(5190)^{a}	931	6 603(2220)^{a}	6 065(2449)^{a}	538	195.0(66.8)^{ab}	200.3(69.0)^{a}	−5.3
Total	30 885	26 447	4438	14 420	12 169	2251	263.7	262.2	1.5
Lower slope									
CWD	15 720(10 913)^{a}	15 596(10 941)^{a}	164	7 502(5195)^{a}	7 424(5208)^{a}	78	24.7(17.1)^{a}	24.5(17.2)a	0.2
Small wood	3560(1210)^{a}	3 231(1245)^{a}	329	1 751(595)^{a}	1 587(612)^{a}	164	18.9(6.1)^{a}	17.4(6.2)^{a}	1.5
Litter	4151(397)^{ab}	4 028(406)^{b}	123	2 047(194)^{ab}	1 993(197)^{b}	54	44.7(4.1)^{ab}	44.2(4.1)^{a}	0.5
Humus	11 038(1414)^{b}	13 410(1559)^{a}	−2373	4 879(591)^{a}	5 935(728)^{a}	−1056	180.2(22.4)^{b}	218.6(26.7)^{a}	−38.4
Total	34 509	36 265	−1757	16 179	16 939	−760	268.5	304.7	−36.2
Watershed average for all components	39 025	34 043	4982	18 197	15 874	2323	357.0	343.0	14.0

Data in parentheses are standard errors.
*in the loss column represents significant differences between pre- and post-burn (paired t-test; $p<0.10$). Differences among slope positions in component pool sizes (pre- and post-analyzed separately) are denoted by letters, where specific forest floor components (e.g. CWD) with the same letter among slope positions are not significantly different (ANOVA and LSD; $p<0.10$).

Table 4
Pre-burn and post-burn C and N concentration (percent)

	C			N		
	Pre-burn	Post-burn	Difference	Pre-burn	Post-burn	Difference
Ridge						
Small wood	49.8(0.52)[a]	33.2(9.8)[a]	16.6	0.50(0.04)[a]	0.38(0.12)[a]	0.12
Litter	50.7(0.33)[a]	39.2(8.6)[a]	11.5	0.97(0.07)[ab]	1.03(0.24)[a]	−0.04
Humus	43.6(2.66)[a]	44.7(2.61)[a]	−1.0	1.45(0.12)[ab]	1.53(0.06)[a]	−0.08
Mid-slope						
Small wood	49.6(0.42)[a]	50.7(0.63)[a]	−1.1	0.59(0.09)[a]	0.61(0.09)[a]	−0.02
Litter	48.6(1.42)[a]	49.8(0.95)[a]	−1.2	0.86(0.03)[b]	1.22(0.12)[a]	−0.36 *
Humus	45.5(1.36)[a]	42.9(2.69)[a]	2.6	1.35(0.08)[a]	1.52(0.10)[a]	−0.17
Lower slope						
Small wood	49.2(0.18)[a]	49.2(0.19)[a]	0	0.55(0.05)[a]	0.56(0.05)[a]	−0.01
Litter	49.3(0.43)[a]	49.5(0.51)[a]	−0.2	1.08(0.02)[a]	1.10(0.02)[a]	−0.02
Humus	44.6(1.21)[a]	44.5(1.17)[a]	0.1	1.64(0.03)[a]	1.64(0.03)[a]	0

Data in parentheses are standard errors.
* in the difference column represents significant differences between pre-burn and post-burn (paired *t*-test; $p<0.10$). Differences among slope positions in component pool sizes (pre- and post- analyzed separately) are denoted by letters, where specific forest floor components (e.g. small wood) with the same letter among slope positions are not significantly different (ANOVA and LSD; $p<0.10$).

Table 5
Mean solute concentrations and pH in the lower section of Wine Spring Creek in western NC, based on weekly samples from February 1993 to November 1996

Solute	Concentrations (mg l^{-1})
NO_3–N	0.04
NH_4–N	−0.005
HCO_3	2.06
PO_4	0.002
Cl	0.46
K	0.34
Na	0.64
Ca	0.54
Mg	0.24
SO_4	1.06
SiO_2	5.06
pH	6.43

example, there was no significant difference ($p<0.10$) between pre- and post-burn regressions of $NO_3^- - N$ concentrations of the sampling station immediately below the burned watershed versus an unburned reference stream sampling station (Fig. 3). Likewise, $NO_3^- - N$ concentrations at the two sites for post-treatment storms were well within the range of values observed for storms during the pre-treatment period (data not shown).

4. Discussion

4.1. Combustion temperatures

The 'stand replacement' burning technique produced maximum combustion temperatures on the ridge position (i.e., >804°C) comparable to those obtained in site preparation burns (630–812°C) on similar sites in the southern Appalachians (Swift et al., 1993). However, one key difference is that the stand replacement burns resulted in a mosaic of intensities, where, for example, one of the ridge plots had combustion temperatures less than 52°C. The lower combustion temperatures in the mid- and lower slope were consistent with the objectives of stimulating oak regeneration and forage production; overstory mortality was less than 3% (Elliott et al., 1997). Extrapolating from temperature tiles as well as visual observations during the fire, we constructed a map of fire intensity across the study area (Fig. 1). In terms of ecosystem restoration, the mosaic of fire intensity/severity obtained in this stand replacement fire both within the ridge position and across the entire slope, may more closely mimic conditions associated with wildfire. This contrasts sharply with the fell and burn technique, which results in much more uniform fire characteristics (Swift et al., 1993).

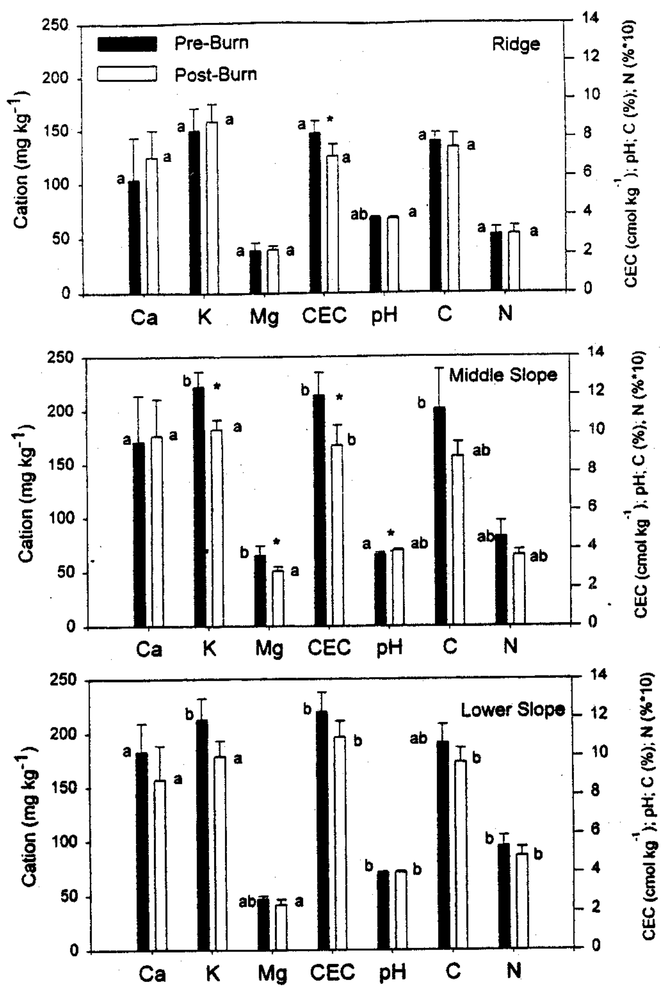

Fig. 2. Soil chemistry responses to stand replacement burning. '*' denotes significant differences between pre- and post-burn (paired t-test; $p<0.10$). Letters compare differences among slope positions (pre- and post- analyzed separately), where soil chemical constituents (e.g. Ca) with the same letters across slope positions do not differ (ANOVA and LSD; $p<0.10$).

Fire intensity has been defined as the upward heat pulse produced by the fire (Ryan and Noste, 1985), and as such, flame temperatures measured by the tiles provide a direct measure of fire intensity. Fire severity incorporates the downward heat pulse and determines (or is defined by) the effects on the forest floor and soil (Wells et al., 1979; Van Lear and Waldrop, 1988). Downward heat pulse (as measured by the temperature tiles) was highest in the ridge, but substantially lower than observed in site preparation burns on similar sites (Swift et al., 1993). In the Swift et al. (1993) study, heat penetrated as much as 45 mm at 60°C and 58 mm at 45°C. Flame temperature may also provide some index of fire severity, as we found significant correlations between flame temperature at both 1 and 2 m height, and percent consumption of CWD, small wood, and litter (Table 6). From a practical standpoint, flame temperatures measured with tiles and heat-sensitive paint may be an easy way to index consumption of some forest floor components. In contrast, there were no relationships between humus consumption and flame temperature, primarily because humus was only slightly consumed in the fire (mean consumption over the entire watershed =10%). In addition, high humus consumption requires backing, smoldering fires with long residence times

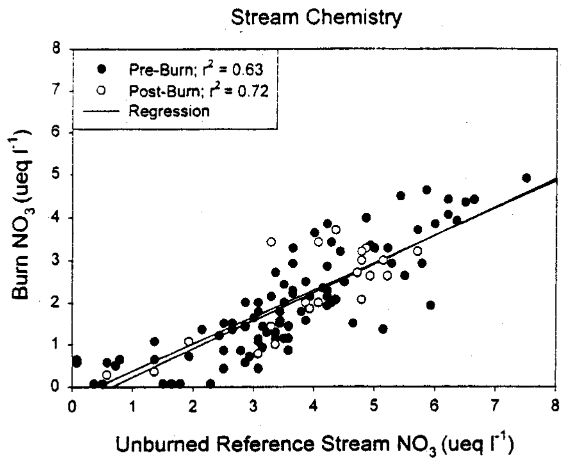

Fig. 3. Stream NO_3 response to stand replacement burning.

which may not be adequately quantified with temperature tiles placed at 1 or 2 m aboveground. There was no relationship between humus consumption and heat penetration (as measured by the forest floor tiles) and only a weak relationship between heat penetration and litter consumption (r^2 values <0.21).

4.2. *Forest floor mass, C, and N losses*

Mass, C, and N losses from the forest floor were a function of the type of material and slope position. On mid- and lower slopes, fire intensity and severity were low and not widespread enough to cause significant consumption and/or volatilization of mass, C, or N. Hence, most of the immediate effects of restoration burning on biogeochemical cycling were confined to the ridge. On the ridge, where the fire was more intense, statistically significant mass, C, and N losses occurred only for the most flammable forest floor components; that is, small wood and litter. Losses from these pools are important from a nutrient cycling standpoint, because they represent potentially available N following decomposition and mineralization (Berg and Staff, 1981; Monk and Day, 1988). Summed across all components in the ridge position, losses were 12 263, 5478, and 78 kg ha^{-1} for mass, C, and N, respectively. In terms of ecosystem N pools, these N losses (i.e. 14% of the aboveground pre-burn total) are relatively small and should rapidly be replenished from increased mineralization, N fixation, and N deposition (Swank, 1984; Adams and Attiwill, 1986; Knoepp and Swank, 1993). In fact, black locust (*Robinia psuedoacacia* L.) density increased 80-fold after burning on the ridge (Elliott et al., 1997), indicating potentially significant N additions via atmospheric N fixation (Boring and Swank, 1984) These results contrast sharply with those found after felling and burning in other pine–hardwood ecosystems (Vose and Swank, 1993). For example, Vose and Swank (1993) found that mass loss exceeded 90 000 kg ha^{-1}, and N losses ranged from 190 to 480 kg N ha^{-1}. The primary factor determining these differences was the large pool of down wood on the fell and burn treatment, due to chainsaw felling of all vegetation prior to burning. This large fuel source increased wood consumption and subsequent C and N losses. Litter mass, C, and N losses were also two- to three-fold greater in the fell and burn treatment (Vose and Swank, 1993) relative to the present study. This was most likely due to the more uniform distribution of high intensity/severity fire on the fell and burn treatments.

Table 6
Regression equations relating percent mass consumption of forest floor components to combustion temperatures determined from heat-sensitive paint applied to ceramic tiles

Tile location	Forest floor component	Equation	r^2	$P>F$
1m aboveground	CWD	0.042+0.0003 (Temp. in °C)	0.40	0.0038
	Small wood	0.251+0.0007 (Temp. in °C)	0.21	0.0228
	Litter	0.088+0.0010 (Temp. in °C)	0.58	0.0001
	Humus	No model		
2m aboveground	CWD	0.045+0.0003 (Temp. in °C)	0.38	0.0049
	Small wood	0.259+0.0007 (Temp. in °C)	0.20	0.0270
	Litter	0.101+0.0011 (Temp. in °C)	0.55	0.0001
	Humus	No model		

4.3. Soil and stream chemistry

Differences in pre- and post-burn soil chemistry imply that the burning treatment had little or no effect on the soil. Other studies have observed increased pH, N, and phosphorus, and base cations (Wells et al., 1979; McKee, 1982; Knoepp and Swank, 1993), but results are often site- and fire-specific. In our study, no soil chemical constituents increased after burning. However, K, Mg, and CEC significantly decreased. We attribute these decreases to temporal variation (March vs. July sampling) in soil chemistry (Yount, 1975; Knoepp and Swank, 1997), which may be greater than the effects of fire on these sites.

With the exception of $NO_3^- - N$, the chemistry of untreated streams draining the WSC Basin are typical of streams in other undisturbed areas of western North Carolina (Swank and Waide, 1988). In contrast, average $NO_3^- - N$ concentrations at the six WSC stream stations (i.e. 0.04 mg l^{-1}) are typical of concentrations in streams draining disturbed watersheds; particularly, those that were clearcut in the past 15 years (Swank, 1988). Several areas within WSC have been logged over the past two decades and thus, the elevated $NO_3^- - N$ may reflect those past disturbances.

The lack of stream $NO_3^- - N$ or $NH_4^+ - N$ response to burning was not unexpected since fire severity was low near the stream and there is no overland flow. Where fires are more severe (i.e. significant consumption of humus) and extensive, small-stream nutrient responses (increased $NO_3^- - N$, $NH_4^+ - N$, PO_4–P, K, Ca, Mg) have been observed (Neary and Currier, 1982; Knoepp and Swank, 1993). In our study, there was no mechanism for long-distance transport of N to the streams. Furthermore, any $NO_3^- - N$ mobilized by burning (Knoepp and Swank, 1993) and transported downslope by subsurface flow would be utilized by vegetation in the unburned or lightly burned riparian and lower slope positions.

5. Conclusions

The impacts of stand restoration burning on the components of the biogeochemical cycle measured in our study are minimal, especially when compared with the alternative treatment of felling and burning. Nitrogen losses were confined to the ridge and were small enough (i.e. 78 kg N ha^{-1}) to be rapidly replenished by atmospheric inputs and N fixation. Soils and streams showed no response and thus, fire effects were limited to the forest floor. Assuming that restoration burning satisfies the silvicultural objectives of pine and oak regeneration, and increased forage production, we conclude that the stand replacement treatment may be preferred over the fell and burn, not only because of lower costs, but also because of significantly lower impacts on biogeochemical cycling (Vose and Swank, 1993; Knoepp and Swank, 1993).

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